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Modifications of Models Predicting Trophic State of Lakes: Adjustment of Models to Account for the Biological Manifestations of Nutrients

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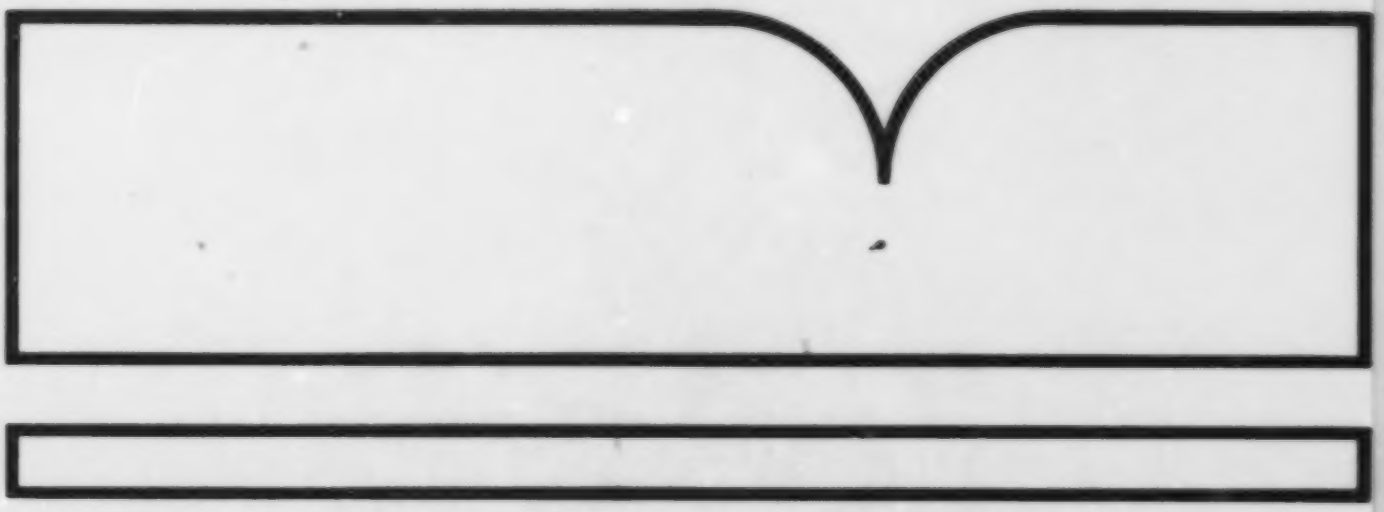

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16. ABSTRACT The strong relationship between total phosphorus and phytoplankton biomass in lakes has been clearly confirmed. What is presently needed in order to predict algal biomass for management decisions on individual lakes is a quantitative understanding of the range in biomass (as measured by chlorophyll a) per unit of phosphorus which extends over orders of magnitude. Data collected from lakes were analyzed determining the environmental factors affecting the response of phytoplankton chlorophyll a to total phosphorus. It was found that light attenuation from other than non-chlorophyll a related interferences can dramatically affect the quantity of phytoplankton biomass in many U.S. lakes. The ratio of biologically available phosphorus to nitrogen is in some cases an important factor in determining the amount of chlorophyll a produced per unit of phosphorus. A method is presented to predict light attenuation due to non-chlorophyll a related interferences. Methods are presented to modify nutrient loading and ambient models which predict the trophic state of lakes to: a) change the trophic classification based on an ambient total phosphorus level to one based on the biological manifestation of nutrients as measured by chlorophyll a, b) determine the critical levels of phosphorus which will result in an unacceptable level of chlorophyll a and, c) account for the unique characteristics of a lake which affect the amount of chlorophyll a produced per unit of phosphorus. It was found that many U.S. lakes would be classified lower (i.e., less eutrophic) if chlorophyll a was used as the trophic classification criteria rather than total phosphorus.

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Adjustment of Models to Account for the Biological Manifestations of Nutrients

by

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LIST OF ABBREVIATIONS

PARAMETER ABBREVIATIONS

CHLA	-- Chlorophyll *a*
DP	-- Dissolved Phosphorus
IN	-- Ammonia-nitrite-nitrate-N
KN	-- Total Kjeldahl-N
N/P	-- Ratio of IN to TP
NO	-- Nitrite-nitrate-N
NH	-- Ammonia-N
OP	-- Dissolved orthophosphorus
RA	-- Response ratio
SD	-- Secchi disk
T	-- Temperature
TP	-- Total phosphorus

MODEL ABBREVIATIONS

A	-- -4.77
AERA	-- the mean summer RA for the lake corrected to what it would be at the 20 µg/l level of TP, i.e., the ambient eutrophic level
AMRA	-- the mean summer RA for the lake corrected to what it would be at the 10 µg/l level of TP, i.e., the ambient mesotrophic level
ATP_{AE}	-- the minimum mean ambient summer lake TP concentration which will cause the lake to become eutrophic corrected to account for the lake's RA

ATP_{AM} -- the minimum mean ambient summer lake TP concentration which will cause the lake to become mesotrophic corrected to account for the lake's RA

ATP_E -- the minimum mean ambient summer lake TP concentration which will cause the lake to become eutrophic

ATP_M -- the minimum mean ambient summer lake TP concentration which will cause the lake to become mesotrophic

B -- -8

ERA -- a constant equal to 0.32 which is the RA predicted from 20 µg/l of ambient TP

ETP -- a constant equal to 20, which is the theoretical minimum ambient concentration of TP in µg/l which will result in eutrophic conditions in a lake and is the level which if not equaled will result in meso- or oligotrophic conditions

HT -- hydraulic residence time

L -- phosphorus areal loading

MRA -- a constant equal to 0.23 which is the RA predicted from 10 µg/l of ambient TP

MTP -- a constant equal to 10, which is the theoretical minimum ambient concentration of TP in µg/l which will result in mesotrophic conditions in a lake and is the level which if not equaled will result in oligotrophic conditions

OCHLA -- observed mean summer CHLA in a lake

ORA -- observed summer ambient RA in the lake

OTP -- observed summer ambient TP in the lake

OTTP -- observed mean tributary TP

OS -- observed SD value for a given lake

P -- hydraulic flushing rate

PS -- predicted SD

$\overline{TP}_{AE}$ -- the minimum mean tributary TP concentrations in µg/l which will cause a lake to be eutrophic at equilibrium corrected to account for the lake's RA

$\overline{TP}_E$ -- the minimum mean tributary TP concentration in µg/l which will cause a lake to be eutrophic

$\overline{TP}_{AM}$ -- the minimum mean tributary TP concentration in µg/l which will cause a lake to be mesotrophic at equilibrium corrected to account for the lake's RA

$\overline{TP}_{M}$ -- the minimum mean tributary TP concentration in µg/l which will cause a lake to be mesotrophic

R -- retention coefficient for phosphorus

RS -- residual SD, an index of non-CHLA-related light attenuation

Z -- mean depth

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INTRODUCTION

Phosphorus supply is considered to be the primary determinant of algal community biomass and production in most temperate zone lakes (Schindler 1974, Williams et al. 1977, Lambou et al. 1976, Dillon and Rigler 1974, Bachmann and Jones 1974, Edmondson 1970). Kalff and Knoechel (1978) state that the strong relationship between total phosphorus and phytoplankton biomass (chlorophyll a) has been clearly confirmed and what is presently needed in order to predict algal biomass for management decisions on individual lakes is a quantitative understanding of the range in biomass (chlorophyll a) per unit of phosphorus which extends over orders of magnitude.

Vollenweider (1968), Dillon (1975), and Larsen and Mercier (1976) have developed models to predict ambient total phosphorus lake concentration from tributary phosphorus loading data. These and other similar models are widely accepted and used to indicate the degree of eutrophy of lakes and evaluate the environmental effects of ambient phosphorus levels in lakes. These models are currently being used to make decisions about lake management, e.g., the manipulation of ambient total phosphorus concentration to produce a desired environmental effect. All of the commonly used models utilize levels of 10 µg/l of ambient phosphorus concentrations to divide lakes into three standard trophic classifications--oligotrophic, mesotrophic, and eutrophic. The assumption is made in the use of these somewhat arbitrary levels of 10 and 20 µg/l of ambient lake phosphorus concentrations that the relationship of phytoplankton biomass to phosphorus is the same for all lakes. However, the utilization and incorporation of phosphorus into phytoplankton biomass varies significantly from lake to lake. The efficiency of utilization of phosphorus is largely dependent upon the availability of light, sufficient supply of other macronutrients (e.g., nitrogen) and micronutrients, and biological availability of the various phosphorus species.

The use of models to predict phytoplankton biomass from actual or potential phosphorus concentrations in individual bodies of water could lead to faulty management decisions if the factors affecting the utilization of phosphorus are not taken into consideration. The purpose of this paper is to evaluate the factors affecting the relationship of phytoplankton biomass to phosphorus levels and to show how to modify models predicting the trophic state of lakes to take these factors into account.

CONCLUSIONS

1) The response ratio, i.e., the amount of chlorophyll *a* produced per unit of total phosphorus, varies greatly in United States lakes, even though there is a strong correlation between total phosphorus and phytoplankton biomass (as measured by chlorophyll *a*) in lakes.

2) Many United States lakes do not reach maximum production of chlorophyll *a* because of interference factors.

3) Light attenuation from non-chlorophyll *a* related interferences can dramatically affect the quantity of phytoplankton biomass present in United States lakes.

4) The ratio of biologically available phosphorus to nitrogen is in some cases an important factor in determining the response ratio in United States lakes.

5) Temperature does not seem to be an important factor influencing the response ratio in United States lakes.

6) Nutrient loading and ambient models which predict the trophic state of lakes can be modified to: a) change the trophic classification based on an ambient total phosphorus level to one based on the biological manifestation of nutrients as measured by chlorophyll *a*, b) determine the "critical levels of total phosphorus which will result in unacceptable levels of chlorophyll *a* so that the level of total phosphorus can be manipulated to achieve the desired use of a given water body and c) account for the unique characteristics of a lake which affects the amount of chlorophyll *a* which will be produced per unit of total phosphorus.

7) Many United States lakes would be classified lower (i.e., less eutrophic) if the biological manifestation of nutrients as measured by chlorphyll *a* is used as the basis of classification rather than total phosphorus.

8) If the manifestations of nutrients, rather than their absolute levels, are considered the primary criteria for beneficial water use, many communities could be spared the unnecessary burden of costly nutrient-removal programs suggested by phosphorus-based trophic classifications.

RECOMMENDATION

It is recommended that loading and ambient models predicting trophic state be adjusted to account for the amount of chlorophyll *a* produced per unit of total phosphorus in a lake because (1) non-chlorophyll *a* light interferences and other interferences in many United States lakes significantly decrease the amount of chlorophyll *a* produced per unit of total phosphorus and (2) excessive algal growth or other manifestations of nutrient enrichment are more important from a water quality standpoint than a trophic classification based on an arbitrary ambient total phosphorus level.

MATERIALS AND METHODS

The data base used to evaluate the factors affecting the relationship of phytoplankton biomass to phosphorus levels came from the 815 lakes (Figure 1) sampled throughout the 48 contiguous United States during the spring, summer, and fall of 1972 through 1975 by the National Eutrophication Survey (NES).

LAKE SELECTION

Lakes included in the NES were selected through discussions with State water pollution agency personnel and U.S. Environmental Protection Agency Regional Offices (U.S. EPA 1975). Screening and selection of lakes sampled in 1972 and 1973 strongly emphasized lakes with actual or potential accelerated eutrophication problems. As a result, the selection was generally limited to lakes:

(1) impacted by one or more municipal sewage treatment plant outfalls either directly into the lake or by discharge to an inlet tributary within approximately 40 kilometers of the lake;

(2) 40 hectares or larger in size, and

(3) with a mean hydraulic retention time of at least 30 days.

Specific selection criteria were waived for some lakes of particular interest to the individual states. During 1974 and 1975, lakes were selected to broaden the spectrum of water quality types represented (U.S. EPA 1975).

FIELD SAMPLING METHODS

Lake sampling was accomplished by two sampling teams, each consisting of a limnologist, pilot, and sampling technician, operating from pontoon-equipped Bell UH-1H helicopters. A mobile field laboratory provided analytical support. With few exceptions, each lake was sampled under spring, summer, and fall conditions. The sampling periods for each season are given by sampling year in Table 1.

Sampling site locations were chosen to define the character of the lake as a whole and to investigate visible or known problem areas, e.g., algal blooms, and sediment or effluent plumes (U.S. EPA 1975). The number of sites was limited by the survey nature of the program and varied in accordance with lake

Figure 1. Number and location of lakes sampled during the years indicated.

TABLE 1. SEASONAL SAMPLING PERIODS FOR 1972 THROUGH 1975

Sampling Year	Number of Lakes	Season		
		Spring	Summer	Fall
1972	231	5/20-7/19	7/20-9/24	9/15-11/16
1973	250	3/7-7/11	7/5-9/18	9/19-11/4
1974	180	3/4-5/12	5/13-9/5	9/11-11/14
1975	154	2/21-6/10	6/11-9/3	9/4-12/11

size, morphological and hydrological complexity and practical considerations of time, flight range, and weather.

The Survey helicopters were equipped with electric winches and approximately 60 meters (m) of hollow-core, multi-conductor cable attached to a submersible pump and an InterOcean Systems'™ sensor package capable of making in situ measurements of conductivity, temperature, optical transmissivity and depth. Rack-mounted equipment located inside the helicopter provided analog recording capabilities and a digital display of the sensor values. An echo sounder, Secchi disk, sample containers and related equipment items necessary for water sampling were also carried in each helicopter (U.S. EPA 1975).

After landing at the station the sensor-pump package was used to collect data and water samples at various depths. Sampling depths were selected to characterize the water column after inspection of analog strip-chart records generated during the initial pass of the sensor package through the water column.

At each sampling depth, water samples were collected for nutrient, alkalinity, pH and dissolved oxygen determinations. For nutrient analyses, two 130 milliliter (ml) polyethylene sample bottles were filled and sent to the Environmental Monitoring and Systems Laboratory-Las Vegas (EMSL-LV) for analysis.

At each sampling station, a chlorophyll a and a phytoplankton sample were taken. These depth-integrated samples were uniform mixtures of water from the surface to a depth of 4.6 m (15 feet) or from the surface to the lower limit of the photic zone, representing 1 percent of surface incident light, whichever was greater. If the depth at the sampling station was less than 4.6 m, the sample was taken from just off the bottom to the surface. Table 2 summarizes sample collection and handling procedures.

TABLE 2. LAKE WATER SAMPLE ANALYSES SUMMARY*

Parameters	Sample Volume	Field Treatment	Where Performed	Depth
Temperature			In situ	Continuous
Chlorophyll a	4-ounce (130 ml)	Refrigerated, dark	Field lab	Photic zone integration
Total phosphorus Total Kjeldahl-N	4-ounce (130 ml)	Unfiltered, $HgCl_2$	EMSL-LV	Select levels
Nitrite-Nitrate-N, Amonia-N	4-ounce (130 ml)	Filtered, $HgCl_2$	EMSL-LV	Select levels

*Modified from U.S. EPA (1975).

A Beckman Electromate™ portable pH meter and combination electrode were used to determine pH as soon as possible following sample delivery to the field laboratory at the end of the day's sampling. Dissolved oxygen samples were titrated with phenylarsine oxide in the mobile field laboratory within 16 hours after collection. Chlorophyll a analyses were performed at the end of each day in the trailer laboratory according to the fluorometric procedure described by Yentsch and Menzel (1963). One of each pair of nutrient samples was filtered through a type HA 0.45-micrometer Millipore membrane filter into a clean, unused polyethylene bottle, recapped, and, along with its unfiltered counterpart, forwarded to EMSL-LV for analysis.

NUTRIENT ANALYSES

The analytical methods utilized to process the samples at EMSL-LV are outlined in Table 3. All of the analyses were performed utilizing adaptations of automated procedures described in Mullins, et al. (1975).

DATA MANAGEMENT

The physical and chemical data were input to STORET (STOrage and RETrieval), the U.S. EPA computer system which processes and maintains water quality data. Data manipulation was as prescribed by Bliss, et al. (1975). Basic calculations for parameters measured in sampled lakes were performed in such a way as to give equal weight to: each depth sampled at a station; each sampling station sampled on an individual lake during a sampling round; and each sampling round on an individual lake during a sampling year. Mean

TABLE 3. ANALYTICAL METHODS AND PRECISION OF LABORATORY ANALYSES

Parameter	Method	Precision
Total Phosphorus	Persulfate oxidation followed by colorimetric determination of antimony-phosphomolybdate complex.	5.0 µg/l P or ± 5%
Nitrite-Nitrate-N	Cadmium reduction followed by the diazotization of sulfanilamide by nitrite coupled with N-(1-naphthyl)-ethylene diamine.	10.0 µg/l N or ± 5%
Ammonia-N	Alkaline phenol hypochlorite reaction producing indophenol blue.	5.0 µg/l N or ± 5%
Kjeldahl-N	Acid digestion followed by the above procedure for ammonia nitrogen.	100.0 µg/l N or ± 5%

parameter values for each sampling station were calculated as follows (Lambou, et al. 1976):

$$\overline{Par_j} = \sum_{i=1}^{D} par_i/D \qquad (1)$$

where: $\overline{Par_j}$ = mean value for a parameter at the j^{th} sampling station during a sampling round (lake-date).

Par_i = value for the i^{th} depth, and

D = the number of depths for which a parameter was measured at the j^{th} sampling station during a sampling round.

Mean parameter values for each sampling round were calculated as follows:

$$\overline{\overline{Par_k}} = \sum_{i=1}^{S} \overline{Par_j}/S \qquad (2)$$

where: $\overline{\overline{Par}}_k$ = mean value for the k^{th} sampling round on a given lake, and

S = number of sampling sites.

Mean lake parameter values for a given sampling year were calculated as follows:

$$\overline{\overline{\overline{Par}}} = \sum_{k=1}^{3} \overline{\overline{Par}}_k/3 \qquad (3)$$

where: $\overline{\overline{\overline{Par}}}$ = mean parameter value for a given sampling year.

Mean parameter values for a given sampling year were calculated only when values were available for the first, second, and third sampling rounds. Equations 1 and 2 were used to determine mean parameter values for total phosphorus (TP), ammonia-N (NH), nitrite-nitrate-N (NO), ammonia-nitrite-nitrate-N (IN), and total Kjeldahl-N (KN), all expressed in micrograms per liter (µg/l); temperature (T) in degrees Celsius (°C), and Secchi disk (SD) in m.

The ratio of IN/TP (N/P) for each sampling station was calculated as follows:

$$\overline{N/P} = \overline{IN}/\overline{TP} \qquad (4)$$

However, at the Equation 1 level a total phosphorus value was deleted if either nitrogen complement was missing. Round and yearly values were calculated using Equations 2 and 3.

Unlike the above parameters measured at various depths, only one chlorophyll a (CHLA) measurement was made at any individual sampling station during a sampling round. Therefore,

$$\overline{CHLA}_j = CHLA \qquad (5)$$

where: $CHLA_j$ = the mean CHLA concentration in micrograms per liter (µg/l) for the j^{th} sampling station during a sampling round, and

CHLA = the CHLA concentration in µg/l for a depth-integrated water sample.

A response ratio (RA) i.e., the amount of CHLA per unit of TP was calculated for each sampling round as follows:

$$\overline{\overline{RA}}_k = \overline{\overline{CHLA}}_k \; / \; \overline{\overline{TP}}_k \qquad (6)$$

where: $\overline{\overline{RA}}_k$ = the response ratio for the k^{th} sampling round on a given lake, while

$\overline{\overline{CHLA}}_k$ and $\overline{\overline{TP}}_k$ = the mean CHLA and TP for the k^{th} sampling round on a given lake.

RELATIONSHIP OF PHYTOPLANKTON BIOMASS TO PHOSPHORUS

A strong relationship between CHLA (a measure of phytoplankton biomass) and TP in lakes has been established by Jones and Bachmann (1976), Dillon and Rigler (1974), Sakamoto (1966), and Carlson (1977). The log-log product moment correlation coefficients reported ranged from 0.85 to 0.98 (Table 4). The inplication of these findings is that phosphorus is the element that controls algal biomass, however, the lakes utilized by the above authors in deriving their relationships are largely phosphorus-limited. No appreciable differences can be detected between the slope of regression equations (summer CHLA regressed on summer or spring TP) from lakes studied in Japan (Sakamoto 1966), Canada (Dillon and Rigler 1974) and North America (Jones and Bachmann 1976 and Carlson 1977). Because of the similarities between the various regression equations derived from lakes studied throughout the world, we feel that they accurately describe the relationship between CHLA and TP under nearly ideal conditions, i.e., without major interferences.

The slopes of the regression lines from the four sources cited above are all greater than 1.4 (Table 4) which indicates that proportional increases in CHLA concentrations accrue at a faster rate than increases in TP concentrations, i.e., the response ratio of CHLA to TP is greater at high concentrations of TP than at low concentrations. This is also shown in Figure 2, where the RA based on Jones and Bachmann's (1976) regression equation is presented. However, the log-log regression equations derived from the lakes we sampled suggest that RA decreases as total phosphorus increases (Table 5). Since lake management decisions to control or manipulate phytoplankton biomass are usually made on summer data and this was the period when we observed the greatest potential response of the phytoplankton community to available nutrients, only summer data are used throughout the remainder of this report.

A histogram of summer RA for 757 lakes and reservoirs located in the conterminous United States is presented in Figure 3. Summer RA varied widely among the various lakes ranging from 0.002 to 2.26 with a mean of 0.29 (Table 5).

Mean summer CHLA is plotted against mean summer TP for the 757 lakes we studied (Figure 4). Superimposed on Figure 4 is the regression equation of Jones and Bachmann (1976). The plots of the vast majority (80 percent) of the 757 lakes shown in Figure 4 lie below the Jones and Bachmann regression line. Of these, most plots occur at TP levels in excess of 20 micrograms per liter. We believe that the reason most lakes do not reach maximum production of CHLA, as described by the regression equation, is because of interference factors.

TABLE 4. REPORTED RELATIONSHIPS BETWEEN CHLA AND TP IN LAKES

Source	Regression Equation*	Number of Lakes	Log-Log Product Moment Correlation Coefficient
Jones and Bachmann (1976)†	Log CHLA = -1.09 + 1.46 Log TP	143	r = 0.95
Dillon and Rigler (1974)‡	Log CHLA = -1.136 + 1.45 Log TP	46	r = 0.95
Sakamoto (1966)‡ (as calculated by Dillon and Rigler)	Log CHLA = -1.13 + 1.58 Log TP	28	r = 0.98
Carlson (1977)†	Log CHLA = -1.06 + 1.45 Log TP	43	r = 0.85
This study†	Log $\overline{\text{CHLA}}$ = -0.11 + 0.64 Log $\overline{\text{TP}}$	757	r = 0.60

*CHLA and TP in µg/l
†Summer CHLA and summer TP
‡Summer CHLA and spring TP

TABLE 5. RA BY SEASON

RA	Season*		
	Spring	Summer	Fall
Mean	0.24	0.29	0.24
Standard deviation	0.27	0.29	0.21
Minimum value	0.001	0.002	0.001
Maximum value	2.81	2.26	1.60
Number of lakes	785	757	777

*A one way ANOVA and Student Newman Keul's multiple range test indicated a significant difference between summer values and fall and spring values at the 0.05 level.

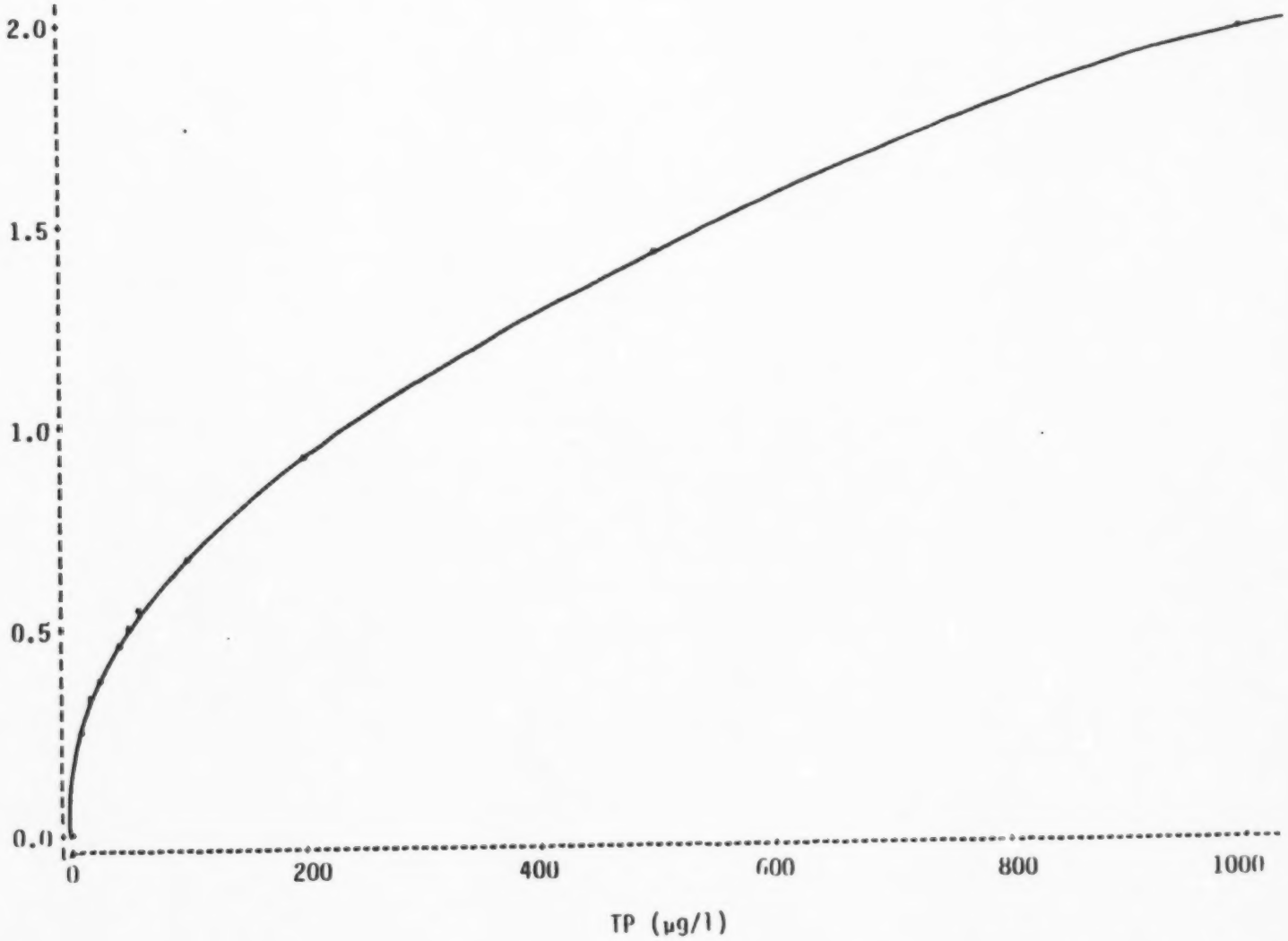

Figure 2. The relationship between summer RA and TP in lakes based on Jones and Bachmann's (1976) regression equation.

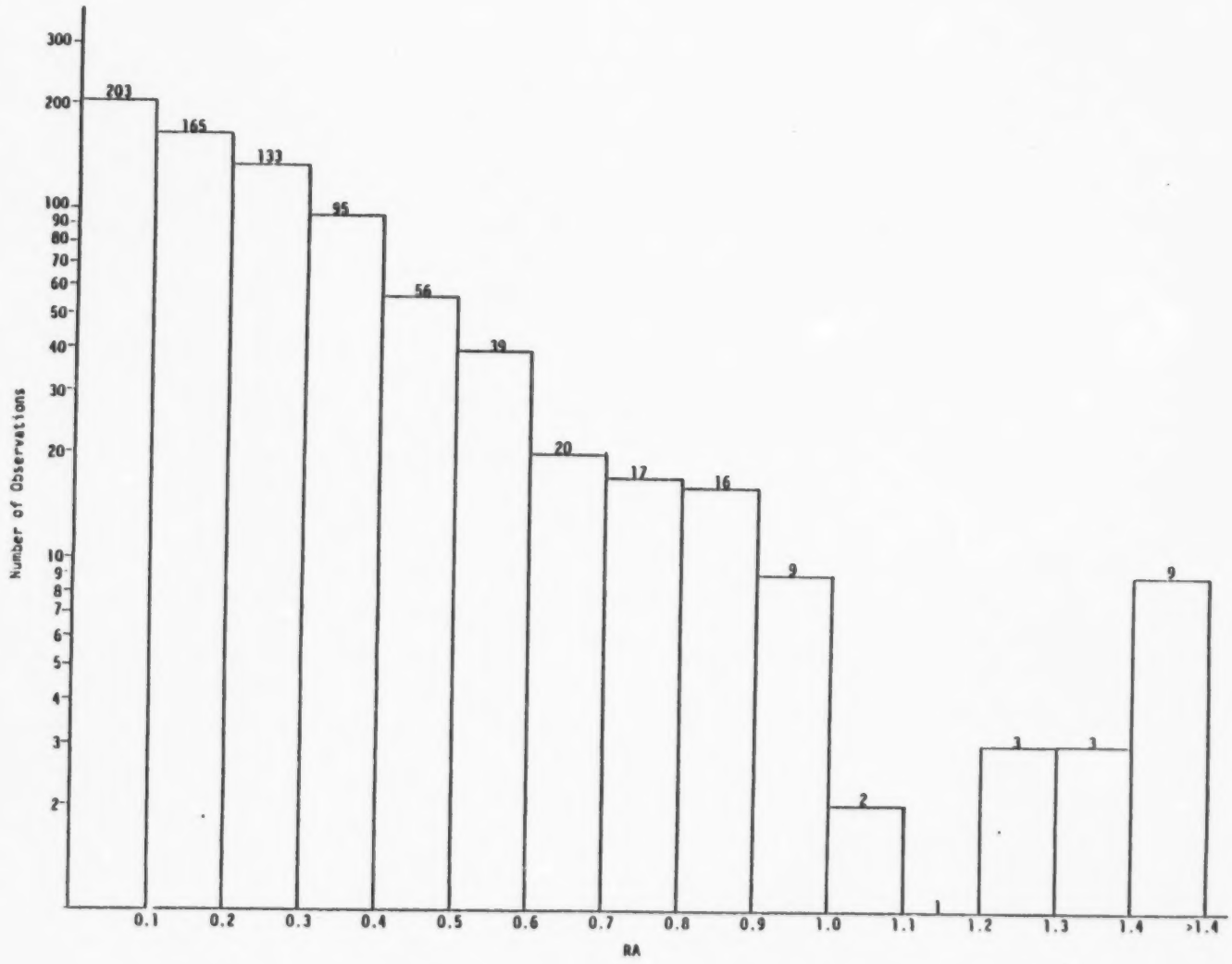

Figure 3. Summer RA for 757 U.S. lakes.

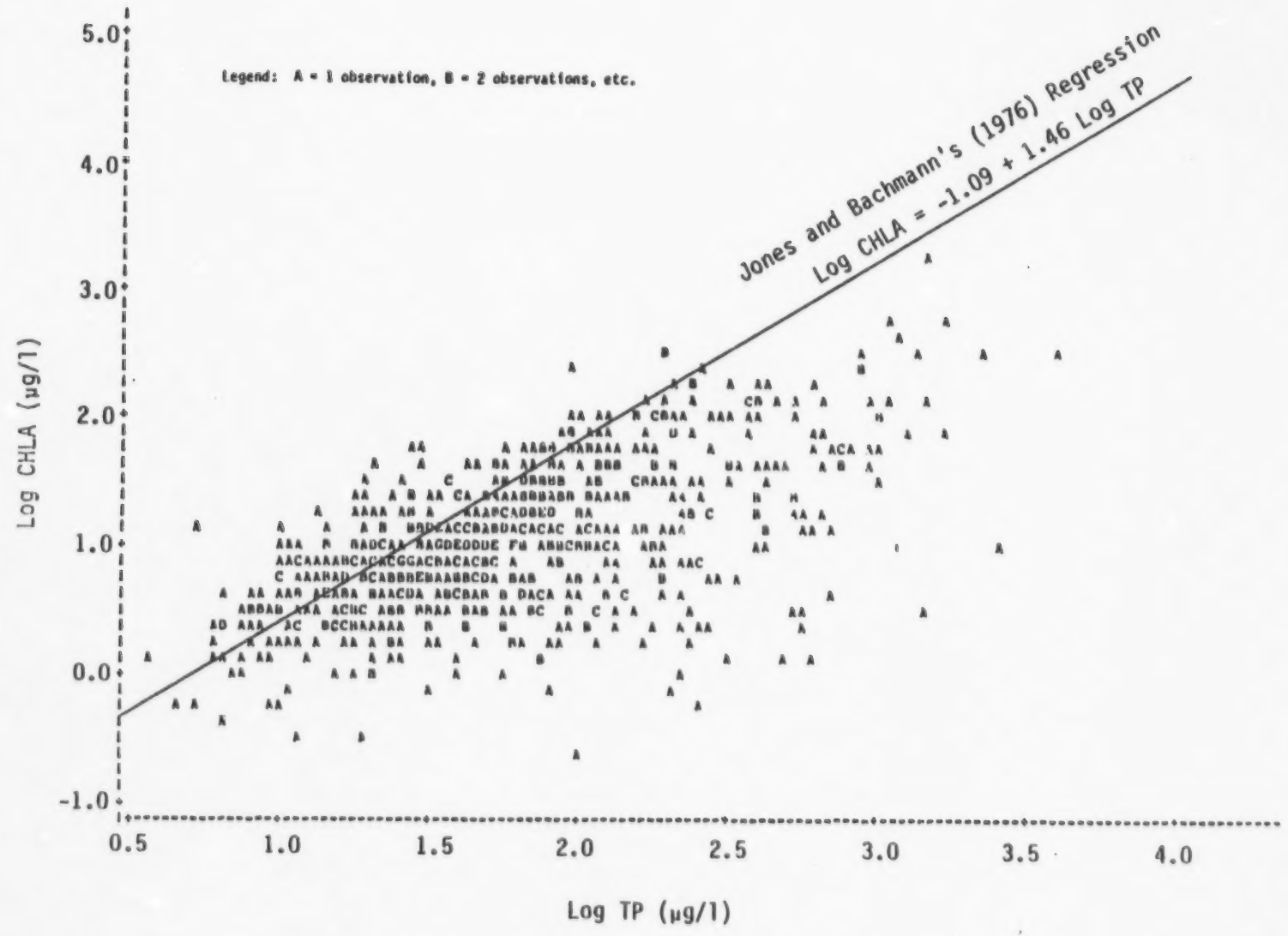

Figure 4. Plot of summer log TP and log CHLA for 757 U.S. lakes.

FACTORS AFFECTING RESPONSE RATIOS

Factors which may prevent phytoplankton CHLA from achieving maximum theoretical concentrations based upon ambient TP levels in a lake include:

1. availability of light;
2. limitation of growth by nutrients other than TP, e.g., nitrogen, carbon, silica, etc.;
3. biological availability of the TP components;
4. domination of the aquatic flora by vascular plants rather than phytoplankton;
5. temperature;
6. short hydraulic retention time; and
7. presence of toxic substances.

LIGHT

The relationship between SD and CHLA concentrations in lakes has been described by a number of investigators (Edmondson 1970, Bachmann and Jones 1974, and Carlson 1977). After analyzing data from 147 lakes, Carlson (1977) described a strong relationship ($r = 0.93$) between CHLA (µg/l) and SD (m), i.e.:

$$\ln SD = 2.04 - (0.68 \ln CHLA) \quad (7)$$

The lakes Carlson utilized were relatively free of non-chlorophyll-related particles. The strength of the relationship Carlson described indicates that light attenuation from any source other than CHLA-related particles was homogenous in the population of lakes represented in his study.

A natural log transformation of our summer CHLA and SD resulted in a correlation coefficient of only 0.56 (Figure 5). This indicates that light attenuation due to non-CHLA-related suspensoids or dissolved color must be an important factor in a number of lakes sampled in our study.

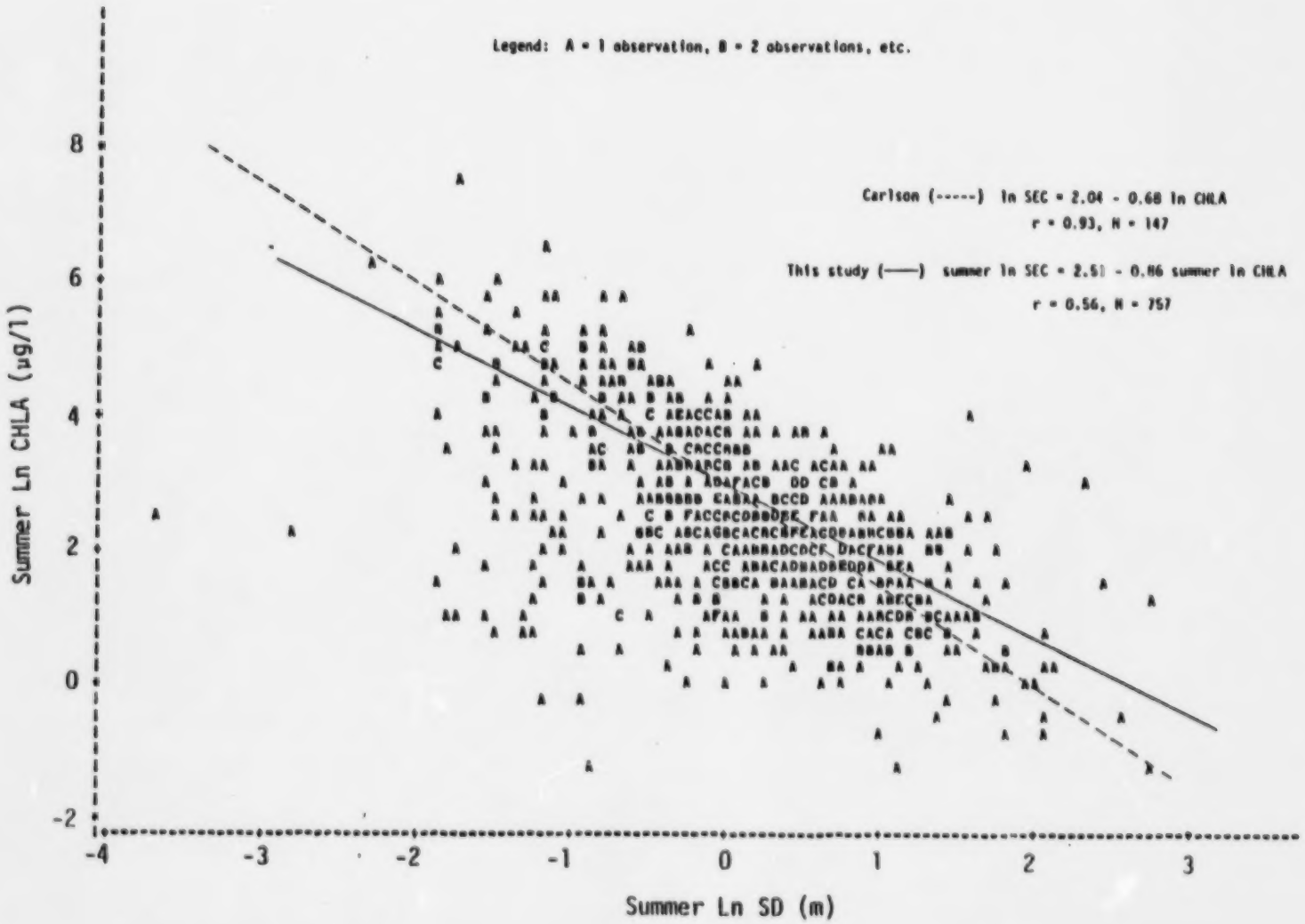

Figure 5. Relationship of summer ln secchi depth to summer ln chlorophyll a for U.S. lakes (Carlson's [1977] regression line is presented for comparison).

If Carlson's regression equation accurately describes the relationship between CHLA and SD in water bodies relatively free of non-CHLA-related particles or dissolved color interferences, it can be utilized in conjunction with observed CHLA concentrations and SD measurements to estimate light attenuation due to non-CHLA-related particles or dissolved color. We did this in the following manner:

$$RS = PS - OS \quad (8)$$

where: RS = residual SD, an index of non-CHLA-related light attenuation;

PS = predicted SD, determined by substituting our mean ambient summer CHLA value for a given lake into Equation 7; and

OS = the observed SD value for a given lake.

If the RS value is positive, non-CHLA-related suspensoids or color interferences are expected. As light attenuation due to the non-CHLA-related factors increases, RS values increase. For a RS value equal to 0, the light attenuation is assumed to be mainly due to CHLA-related particles. We calculated a mean summer RS value for each of our lakes by the method presented above and plotted the values against the RA to examine the effects of light interference upon the relationship of CHLA to TP (Figure 6).

Theoretically, if all of the light attenuation was due to CHLA-related particles, then, according to Carlson's equation (Equation 7), no lake would have a RS value below zero. Obviously, all lakes, including those investigated by Carlson, have some light attenuation due to non-chlorophyll-related particles and color interferences. Therefore, the true zero line (where virtually all light interferences are due solely to CHLA-related particles) is less than a RS value of zero. The true zero line should contain lakes that extend over the full range of RA values, since many lakes are relatively free of non-CHLA-related particles. An examination of Figure 6 and a consideration of experimental error led us to the conclusion that the population of lakes which extends over the full range of RA and lies between RS values of 0.46 to -2.00 encompasses the true zero line. Since SD values less than true zero are theoretically impossible, we believe the RS values less than -2.00 in Figure 6 are due to experimental error, and are not used in our analysis of the factors affecting RA's. With the above criteria, the lakes can be divided into two groups by their RS values (Figure 6). One group has RS values of >0.46 (high non-CHLA light interference) while the other group has RS values between 0.46 and -2.00 (low non-CHLA light interference).

We divided the lakes into 3 RA groups: "very low" (0.00 to <0.50), "low" (0.50 to <1.0), and "high" ($\geq$1.0) (Figure 6). Our rationale for this classification is that with a response of approximately 1.0 or above, all the TP in the system is "theoretically" associated with cellular CHLA (Strickland 1960). Therefore, at a RA of 1.0 or greater, the algae are utilizing all available phosphorus and incorporating it into cellular material. At a RA

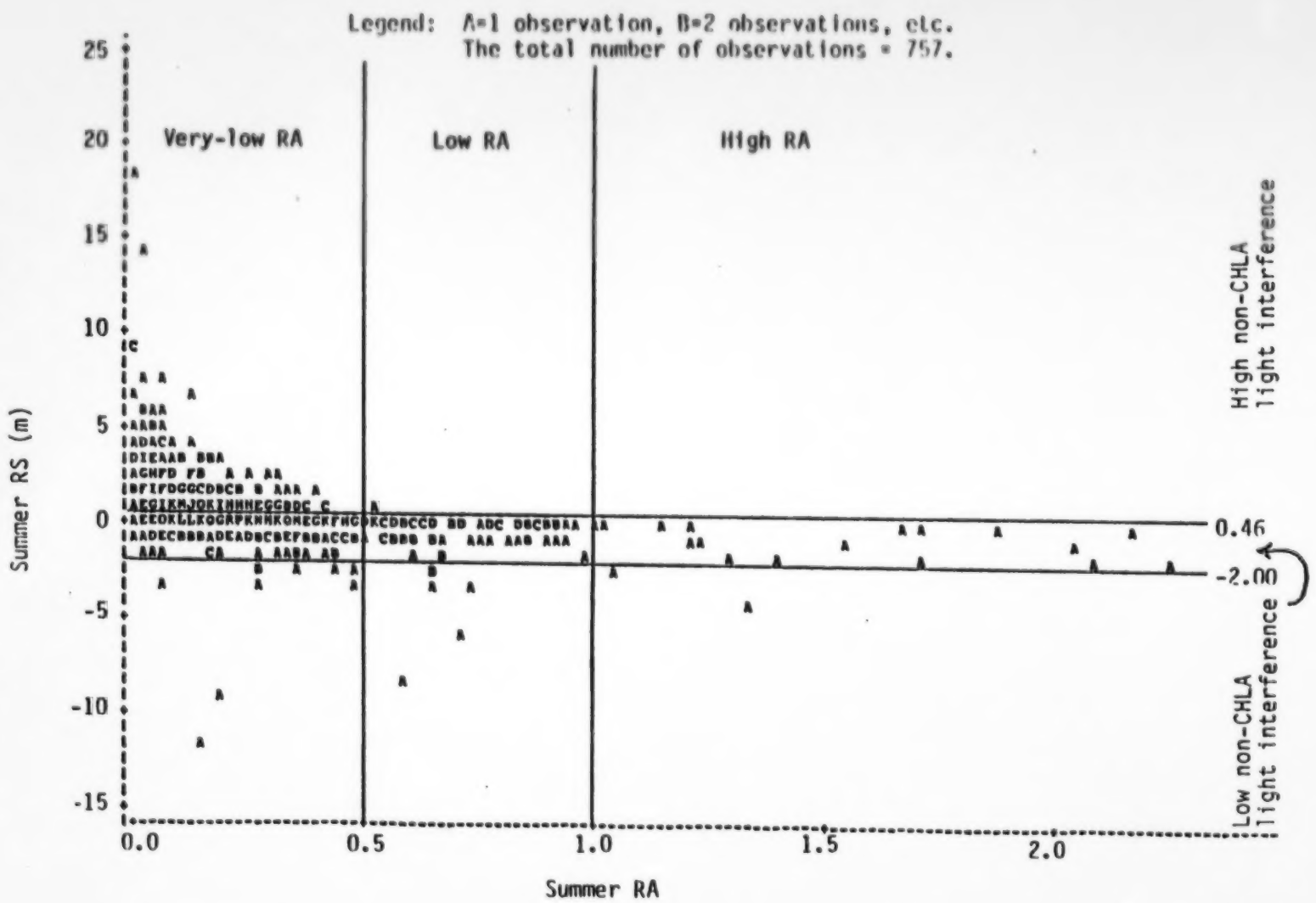

Figure 6. The relationship of summer RS to RA.

of 0.5, half of the TP is theoretically tied up in phytoplankton cells, while the other half of the TP pool is extracellular. These divisions are somewhat arbitrary, as the relationship between cellular phosphorus and CHLA can dramatically change, depending upon environmental conditions and phytoplankton community composition, e.g., luxury uptake by selected phytoplankton forms and increased CHLA production per unit of biomass under low-light conditions. Even though the divisions of the response ratio are somewhat arbitrary, they provide a useful mechanism to compare major response groups.

Only 1 of 104 water bodies was found in the low or high response groups which had a RS >0.46 m while 287 of 632 (45%) of the water bodies with a very-low response had a RS >0.46 m. The low non-CHLA light interference, very-low RA group had approximately double the mean TP concentration but seven times the CHLA concentration as compared to the high non-CHLA light interference group, very-low RA group (Table 6). This suggests that non-CHLA light interference is an important factor in controlling the RA. The low non-CHLA light interference very-low RA group had the highest mean TP concentration (208 µg/l). The RS in the low non-CHLA light interference group decreases from -0.19 for the very-low RA group to -0.35 and -0.64 for the low and high RA groups, respectively (Table 6).

TABLE 6. MEAN SUMMER PARAMETER VALUES BY NON-CHLA LIGHT INTERFERENCE AND RA GROUPS

Non-CHLA* Light Interference	RA†	N‡	TP (µg/l)	KN (µg/l)	CHLA (µg/l)	T °C	RA	RS (m)
	Very-Low	345	208	1234	35.8	22.0	0.24	-0.19
LOW	Low	88	75	1286	50.4	23.2	0.69	-0.35
	High	15	161	2003	216.4	24.0	1.59	-0.64
HIGH	Very-Low	287	96	637	5.6	21.1	0.12	2.00

*Low non-CHLA light interference = -2.00 m RS $\leq$0.46 m, while high non-CHLA light interference = >0.46 m RS.
†Very-low response = <0.5 RA, low response = 0.5 $\geq$ RA $\leq$1.0, and high response = >1.0 RA.
‡N - the number of lakes per response and light interference groups. However, the number of observations may be slightly less for a given parameter due to missing values.

Mean CHLA concentrations in the various non-CHLA light interference groups seem to increase with increasing mean KN levels (Table 6). One exception to this occurred in the high non-CHLA light interference, very-low RA group where the mean CHLA level was only 5.6 µg/l but the corresponding KN concentration was relatively high (637 µg/l). This indicates that a large percentage of the nitrogen present in this group is not associated with phytoplankton. This same group had the highest amount of non-CHLA-related light interference as is evident from its high RS (2.00). The nitrogen not associated with phytoplankton is most likely caused by high levels of allochthonous organic materials or resident bacterial or zooplankton populations in these lakes.

It is concluded that in many United States lakes, light attenuation from other than CHLA-related interferences can dramatically affect the quantity of phytoplankton biomass present.

NITROGEN TO PHOSPHORUS RATIOS

If a nutrient besides phosphorus is limiting phytoplankton growth, a low RA should result. Nitrogen and phosphorus are frequently mentioned as the nutrients most likely to limit growth of plants. The concept of a limiting nutrient, as related to Liebig's Law of the Minimum, is that some nutrient, least available relative to the growth requirements of a given organism, imposes primary limitation on the growth of that organism.

Various estimates have been proposed as to what constitutes that ratio of nitrogen to phosphorus at which the addition of either results in the limitation by the other. Such estimates have been reported as low as 5/1 and as high as 30/1 or more, by weight, generally centering about 12/1 to 14/1. These are in reasonable agreement with theoretical needs based upon stoichiometric equations of algal constituents (Vollenweider 1968).

While the limiting nutrient concept has some utility in allowing prediction of the potential growth limits of laboratory monocultures, its extrapolation into natural system studies should be approached warily. A number of enrichment studies have found the effects of phosphorus and nitrogen to be interdependent (Hutchinson 1941; Goldman and Armstrong 1969; Ketchum 1939; Powers et al. 1972), influenced by the presence of trace organic materials in the waters (Rodhe 1958), and dependent upon previous algal culture exposure, i.e., prior luxury uptake of nitrogen or phosphorus.

It is not unlikely that a mixed natural phytoplankton population would contain taxa with a range of optimal growth requirements and predisposing nutritional status. Addition of either nitrogen or phosphorus could potentially evoke a net increase in phytoplankton growth, especially in those cases in which the ambient nitrogen/phosphorus ratio is intermediate between the optimal growth requirements of the various phytoplankton elements. Utilization of nitrogen/phosphorus ratios does not consider the possibilities of other limitations, e.g., carbon, trace elements (iron, manganese, copper, zinc, molybdenum, chlorine, cobalt, boron, silicon, sodium, vanadium and iodine), co-factors, and vitamins. The presence of organic materials may increase nutrient assimilation in some members of the community (Goldman

and Armstrong 1969; Rodhe 1958), or inhibit it in others (Williams 1975). Conditions may exist, over a range of nitrogen/phosphorus values, favoring response to the addition of either phosphorus or nitrogen and representing essentially "net co-limitation." Different species within a community may be limited by different nutrients simultaneously (Fitzgerald 1964). A theoretical basis for simultaneous co-regulation of the specific growth rate of a single population by multiple nutrient is presented by Sykes (1974).

Lambou et al. (1976) divided lakes into phosphorus-limited [IN/dissolved phosphorus (DP) >14], transition (10<IN/DP≥14), and nitrogen-limited (IN/DP<10) groups based upon the yearly mean IN/DP observed in the lake. They state that, while admittedly arbitrary, the suggested divisions represent a convenient means of comparing groups of lakes presumably representing "largely nitrogen-limited" and largely "phosphorus-limited" populations, and a third group representing a buffer between the first two groups. Their groupings were determined by comparing the ratios of IN to dissolved orthophosphorus (OP) in lake water upon which the limiting nutrient was determined by the Algal Assay Procedure Bottle Test. This test (EPA 1971) utilizes the response of the green alga Selenastrum capricornutum to nutrient spikes, usually nitrogen and phosphorus, alone and in concert, to determine the growth-limiting nutrient.

In our analysis of the lakes sampled by NES, because OP or DP was not available for all of the lakes sampled, we used TP instead of DP in calculating N/P ratios for dividing the lakes into phosphorus-limited (IN/TP>14), transitional (10<IN/TP≥14), and nitrogen-limited (IN/TP<10) groups. Because OP and DP must be equal to or less than TP, this would make N/P ratios reported here somewhat smaller than those we have reported previously. However, we do not believe that this invalidates the comparisons to show relative trends.

Summer N/P ratios for 736 lakes classified 552 (75%) of them as nitrogen-limited, 62 (8%) as transitional and 122 (17%) as phosphorus-limited (Table 7). Even though 75 percent of the lakes were classified as nitrogen-limited, this is somewhat misleading, as many of these are nitrogen-limited by the criteria used here only because of the extremely high summer ambient TP levels. If the summer ambient level of TP was reduced in these lakes to 20 µg/l (a level considered by many to be high enough to evoke eutrophic conditions), approximately 50 percent of the 552 lakes originally classified as nitrogen-limited would be classified in the phosphorus-limited or transitional categories.

The number of lakes classified according to summer N/P ratios as nitrogen-limited, transitional, and phosphorus-limited in the various non-CHLA light interference and RA groups is given in Table 7. Only 70 percent of the high non-CHLA light interference, very-low RA group were classified as N-limited (median N/P=5.3) as compared to 81 percent of the low non-CHLA light interference, very-low RA groups (median N/P=3.9). This suggests that the part of the reduced response of the low non-CHLA light interference, very-low RA group is due to the extremely low N/P ratios. Further evidence of this is that within the low non-CHLA light interference group, the medium N/P increases from 3.9 in the very-low RA group to 4.6 in the low RA group and

TABLE 7. NUMBER AND PERCENT OF LAKES CLASSIFIED ACCORDING TO SUMMER N/P RATIOS AS NITROGEN-LIMITED, TRANSITIONAL, AND PHOSPHORUS-LIMITED BY NON-CHLA LIGHT INTERFERENCE AND RA GROUP

	NON-CHLA LIGHT INTERFERENCE														
	LOW					HIGH					TOTAL				
RA*	Num[K]	N-limit[T]	Tran[T]	P-limit[T]	Med N/P[C]	Num[K]	N-limit[T]	Tran[T]	P-limit[T]	Med N/P[C]	Num[K]	N-limit[T]	Tran[T]	P-limit[T]	Med N/P[C]
very-low	345	278(81%)	21(6%)	46(13%)	3.9	287	201(70%)	32(11%)	54(19%)	5.3	632	479(76%)	53(8%)	100(16%)	4.5
low	88	64(73%)	6(7%)	18(20%)	4.6	1	1(100%)	--	--	5.6	89	65(73%)	6(77%)	18(20%)	4.7
high	15	8(53%)	3(20%)	4(27%)	10.1	--	--	--	--		15	8(53%)	3(20%)	4(27%)	10.1
TOTAL	448	350(78%)	30(7%)	68(15%)	4.7	288	202(70%)	32(11%)	54(19%)	5.4	736	552(75%)	62(8%)	122(17%)	4.6

*very-low = RA <0.5; low = 0.5 ≥ RA <1.0; high = RA ≥1.0

[K]num = number of lakes for which summer N/P ratios were available.

[T]N-limit = number of nitrogen-limited lakes; Tran = number of transitional lakes; P-limit = number of phosphorus-limited lakes; the percent of the lakes in each category is given in parentheses.

[C]Med N/P = the median N/P for all of the lakes in the category.

then to 10.1 in the high RA group (Table 7). The ratio of biologically available phosphorus to nitrogen forms, as theoretically represented by N/P ratios, is in some cases an important factor in determining the RA ratios in lakes.

TEMPERATURE

Solar radiation is required for photosynthesis and heat maintenance in the environment. In general, approximately 50 percent of the total incident light is transformed into heat and is extinguished in the first meter of water (Reid 1961). Van't Hoff's principle states that the rate at which biological processes proceed is increased nearly two-fold with each 10°C rise in temperature. The principle operates only within the range of tolerance of a given species, and is further restricted by an optimal point within the over-all range. Temperature is important to freshwater algae in determining general geographical distribution and control rates of anabolic and catabolic fluxes, as well as rates of the nutrient reflux, and it may affect productivity by accelerating or retarding physiological processes (Smith 1950, Mortimer 1969). However in our analysis of the data collected from the lakes sampled by NES, we were not able to demonstrate any significant seasonal relationships between T and RA (Table 8). The log-log product moment correlation coefficients ranged from only 0.08 to 0.18. The implication of these findings is that T is not an important factor influencing RA over the range of temperature values (2.6 to 32.2°C) observed in the lakes.

TABLE 8. THE RELATIONSHIP BETWEEN LOG T AND LOG RA BY SEASON IN U.S. LAKES

	Season		
	Spring	Summer	Fall
Product moment correlation coefficients	0.11	0.18	0.08
Number of observations	769	754	747

MODIFICATION OF MODELS TO PREDICT TROPHIC STATE

LOADING MODELS

The phosphorus loading -- mean depth -- relationship formulated by Vollenweider (1968) has been widely accepted and used to indicate the degree of eutrophy of lakes and evaluate the level of phosphorus loading to lakes. Dillon (1975) pointed out that too little thought and criticism was given to the limitations of the model by people using it. Subsequently, modifications of the basic mass balance equation have been derived to predict mean ambient lake phosphorus concentrations at equilibrium. Dillon (1975) utilizes phosphorus areal loading (L), the retention coefficient for phosphorus (R), the hydraulic flushing rate (P), and mean depth (Z) in a plot of the form

$$\frac{L\,(1 - R)}{P} \text{ versis } Z \qquad (9)$$

to estimate trophic state. Vollenweider (1975) revised his original formula to include HT, hydraulic residence time, so that areal phosphorus loading (L) is plotted against mean depth (Z) divided by HT. Larsen and Mercier (1976) provide an alternative (to prior loading concepts) which avoids the criticism of Edmondson (1970) that the effect of an increasing phosphorus load upon a lake depends, in part, upon whether that increase results from increases in influent flows, concentrations, or both. The Larsen/Mercier formula plots mean tributary TP concentration against the phosphorus retention coefficient, which they called R experimental, computed in the same way as Dillon's R, i.e.,

$$R = 1 - \frac{\text{TP leaving system}}{\text{TP entering system}} \qquad (10)$$

It is interesting to note that in applying the above-mentioned formulas, each of the authors selected to use levels of 10 and 20 µg/l of ambient lake TP to divide lakes into the three standard trophic classifications -- oligotrophic, mesotrophic, and eutrophic.

The most important public concern involves the manifestation of nutrients, such as excessive algal growth, rather than phosphorus levels. However, "critical" levels of total phosphorus need to be determined, so that meaningful phosphorus control procedures can be accomplished on a given lake.

The theoretical relationship between the summer RA and TP based upon Jones and Bachmann's (1976) regression equation has already been presented in Figure 2. A log-log transformation of the RA and TP concentration results in a straight line function (Figure 7). This straight line function can be used to provide a basis of comparison between the theoretical RA and the actual RA for a given lake at a given TP level so that loading models can be modified to:

1. change the trophic classification based on an ambient TP level to one based on the biological manifestation of nutrients as measured by CHLA,

2. determine the "critical" levels of TP which will result in an unacceptable level of CHLA concentration so that the level of TP can be manipulated to achieve the desired use of a given water body, and

3. account for the unique characteristics of a lake or reservoir which effect the RA.

The Larsen and Mercier (1976) model predicts the mean tributary TP concentration which would cause eutrophic or mesotrophic conditions as follows:

$$\overline{TP}_E = \frac{ETP}{1-R}, \quad \text{or} \tag{11}$$

$$\overline{TP}_M = \frac{MTP}{1-R} \tag{12}$$

where: $\overline{TP}_E$ = the minimum mean tributary TP concentration in µg/l which will cause a lake to be eutrophic at equilibrium,

$\overline{TP}_M$ = the minimum mean tributary TP concentrations in µg/l which will cause a lake to be mesotrophic at equilibrium,

ETP = a constant equal to 20, which is the theoretical minimum ambient µg/l of TP in a lake resulting in eutrophic conditions and is the level which if not equaled or exceeded will result in meso- or oligotrophic conditions, and

MTP = a constant equal to 10, which is the theoretical minimum ambient µg/l of TP in a lake resulting in mesotrophic conditions and is the level which if not equaled or exceeded will result in oligotrophic conditions.

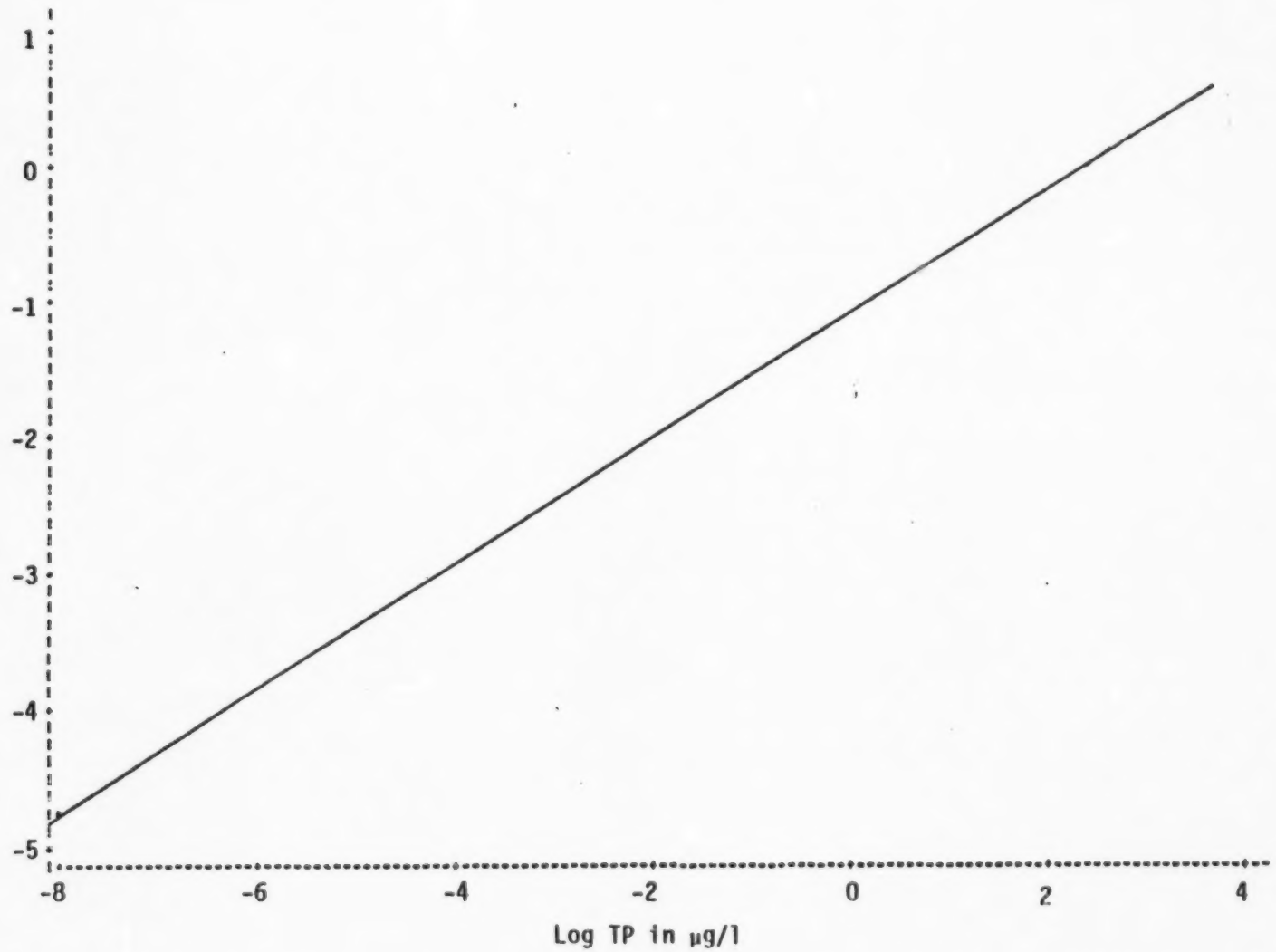

Figure 7. The relationship between summer log RA and log TP based on Jones and Bachmann's (1976) regression equation.

The Larsen and Mercier equations (i.e., Equations 11 and 12) can be corrected to account for the RA of a specific lake as follows:

$$\overline{TP}_{AE} = \frac{ETP(ERA/AERA)}{1-R}, \quad (13)$$

$$\overline{TP}_{AM} = \frac{MTP(MRA/AMRA)}{1-R} \quad (14)$$

where: $\overline{TP}_{AE}$ = the minimum mean tributary TP concentrations in µg/l which will cause a lake to be eutrophic at equilibrium corrected to account for the lake's RA,

$\overline{TP}_{AM}$ = the minimum mean tributary TP concentrations in µg/l which will cause a lake to be mesotrophic at equilibrium corrected to account for the lake's RA,

ERA = a constant equal to 0.32 which is the RA predicted from 20 µg/l of ambient TP utilizing Jones and Bachmann's (1976) regression equation,

MRA = a constant equal to 0.23 which is the RA predicted from 10 µg/l of ambient TP utilizing the Jones and Bachmann's (1976) regression equation,

AERA = the mean summer RA for the lake corrected to what it would be at the 20 µg/l level of TP i.e., the ambient eutrophic level, and

AMRA = the mean summer RA for the lake corrected to what it would be at the 10 µg/l level of TP, i.e., the ambient mesotrophic level.

The ERA constant of 0.32 was determined from utilizing the ETP constant of 20 µg/l of ambient TP in the Jones and Bachmann (1976) regression equation given in Table 4 and which is as follows:

$$\log \text{ µg/l CHLA} = -1.09 + 1.46 \log \text{ µg/l TP} \quad (15)$$

Substituting in 20 µg/l for TP, log CHLA is equal to 0.81 and CHLA is equal to 6.4. Therefore the ERA is equal to 6.4/20 or 0.32. Similarly, the MRA constant of 0.23 was determined utilizing the MTP constant of 10 µg/l of ambient TP.

The AERA is determined from the following equation:

$$\log AERA = \left[\frac{\log ORA - A}{\log OTP - B}\right]\left[\log ETP - B\right] + A \qquad (16)$$

where: ORA = the observed summer ambient RA in the lake,

OTP = the observed summer ambient TP in the lake,

A = -4.77 which is the log of the RA determined from Equation 15 utilizing a TP concentration at approximately 0 (since log 0 is undefined, an extremely low TP concentration, i.e., 0.00000001 µg/l, was used to approximate 0 on the log scale), and

B = -8 which is the log of the TP, i.e., 0.00000001 µg/l, which is used to approximate 0 in Equation 15.

Substituting into Equation 16,

$$\log AERA = \left[\frac{\log ORA + 4.77}{\log OTP + 8}\right] 9.30 - 4.77 \qquad (17)$$

The AMRA is determined from the following equation:

$$\log AMRA = \left[\frac{\log ORA - A}{\log OTP - B}\right] \log MTP - B + A \qquad (18)$$

Substituting into Equation 18,

$$\log AMRA = \left[\frac{\log ORA + 4.77}{\log OTP + 8}\right] 9 - 4.77 \qquad (19)$$

The constants used in Equations 16 and 18 are used to establish the slope of a line (Figure 7) which begins at -4.77 (log RA) and -8 (log TP). Using the ORA and the OTP the RA is adjusted using the relationship shown in Figure 7, which was determined from the Jones and Bachmann (1976) regression equation (Equation 15) to one which would cause eutrophic (AERA) or mesotrophic conditions in the lake (AMRA).

A comparison of using the Larsen and Mercier equations (Equations 11 and 12) to predict trophic state with the modified equations to account for a

lake's RA (Equations 13 and 14) can be made using the data we collected from a lake during 1973. Our data showed that the lake had a:

$$\text{OTP} = 36.3\ \mu g/l,$$

$$\text{observed mean summer CHLA (OCHLA)} = 6.3\ \mu g/l,$$

$$1\text{-}R = 0.71,$$

$$\text{ORA} = 0.17, \text{ and}$$

$$\text{observed mean tributary TP (OTTP)} = 57.3\ \mu g/l.$$

Substituting into Equation 11 we find:

$$\overline{TP}_E = \frac{20}{0.71} = 28.2\ \mu g/l \tag{20}$$

Since 28.2 µg/l of TP represent the theoretical mean tributary concentration which will cause the lake to be eutrophic under steady state conditions and the OTTP is 57.3 µg/l, the use of Equation 11 would classify the lake as eutrophic. Substituting into Equation 13:

$$\overline{TP}_{AE} = \frac{20(0.32/0.13)}{0.71} = 69.3\ \mu g/l \tag{21}$$

Since 69.3 µg/l is greater than 57.3 µg/l, we find if we use the modified equation which accounts for the lake's RA, the lake would be classified as eutrophic and could possibly be either mesotrophic or oligotrophic. To determine whether it is mesotrophic or oligotrophic, we substitute into Equation 14 as follows:

$$\overline{TP}_{AM} = \frac{10(0.23/0.10)}{0.71} = 32.4\ \mu g/l \tag{22}$$

Since 32.4 µg/l is less than 57.3 µg/l we would classify the lake as mesotrophic.

AMBIENT MODELS

As with loading models, ambient models can be adjusted to account for the observed RA in a lake. Using the commonly accepted convention of 10 and 20 µg/l of ambient lake TP to divide lakes into three standard classifications -- oligotrophic, mesotrophic, and eutrophic, we find that:

$$ATP_E = 20 \text{ µg/l, and} \quad (23)$$

$$ATP_M = 10 \text{ µg/l} \quad (24)$$

where: ATP_E = the minimum mean ambient summer lake TP concentration which will cause the lake to become eutrophic and

ATP_M = the minimum mean ambient summer lake TP concentration which will cause the lake to become mesotrophic.

Equations 23 and 24 can be corrected to account for the RA of a specific lake as follows:

$$ATP_{AE} = ETP(ERA/AERA), \quad (25)$$

$$ATP_{AM} = MTP(MRA/AMRA), \quad (26)$$

where: ATP_{AE} = the minimum mean ambient summer lake TP concentration which will cause the lake to become eutrophic corrected to account for the lake's RA and

ATP_{AM} = the minimum mean ambient summer lake TP concentration which will cause the lake to become mesotrophic corrected to account for the lake's RA.

Using the same data collected from a lake during 1973 which we previously used as an example of adjusting loading models, we find that:

$$ATP_{AE} = 20(0.32/0.13) = 49.2 \text{ µg/l, and} \quad (27)$$

$$ATP_{AM} = 10(0.23/0.10) = 23.0 \text{ µg/l}. \quad (28)$$

Since the OTP was 36.3 µg/l, which is between the ATP_{AE} and ATP_{AM} we would classify the lake as mesotrophic using the correction for its RA; however, using the normal convention of 20 µg/l of ambient lake TP to determine the eutrophic dividing line, we would have classified the lake as eutrophic.

RESULTS FROM USING MODIFIED MODELS

Equations 13, 14, 27, and 28 define trophic levels by the biological manifestation of nutrients as measured by CHLA levels, rather than by TP levels. Substituting 10 and 20 µg/l of TP (the normally used dividing values between oligotrophic, mesotrophic, and eutrophic) into Equation 15 [the Jones and Bachmann (1976) regression equation to predict CHLA from TP], we arrive at levels of 2.3 and 6.4 µg/l of CHLA as the dividing values between oligotrophic, mesotrophic, and eutrophic.

Trophic classifications based on mean ambient summer TP and CHLA levels of 757 lakes for which we had data are presented in Table 9 and Figure 8. The TP criteria subdivided the lakes into the following trophic categories: 59 oligotrophic, 97 mesotrophic, and 621 eutrophic, while the CHLA criteria subdivided the lakes into the following trophic levels: 78 oligotrophic, 197 mesotrophic, and 482 eutrophic (Table 9). Both trophic classification schemes classified 19 water bodies as oligotrophic, 45 as mesotrophic, and 447 as eutrophic, for a total agreement of 511 or 68 percent. Of the 246 water bodies which were classified differently by the two schemes, 193 (25%) are classified lower (less eutrophic), while only 53 (7%) are classified as higher (more eutrophic) by the CHLA criteria than by the TP criteria (Table 9). If the large population of lakes used for this comparison is, in fact, representative of conditions throughout the United States, it can be expected that some 25% of the nation's lakes would be classifed lower (less eutrophic) using CHLA levels as a trophic classification criteria. And, if the manifestations of nutrients rather than their absolute levels are considered the primary criteria for beneficial water use, many communities may be spared the unnecessary burden of costly nutrient-removal programs suggested by phosphorus-based trophic classifications. Also, it should be emphasized that eutrophication does not necessarily result in the degradation of water quality. Whether or not the process adversely impacts water quality depends on the beneficial use for which a lake is being managed and the degree of nutrient enrichment.

The Vollenweider, Dillon, and Larsen/Mercier loading models for predicting ambient lake phosphorus concentrations and classifying lakes by trophic state have been compared by Hern et al. (1978). The Larsen/Mercier and Dillon models gave comparable results in ranking 39 lakes relative to known ambient phosphorus concentrations. The Vollenweider model, which does not have a phosphorus retention time capacity component, was unable to achieve the high rank correlations found with the other models. The Larsen/Mercier model was superior to the Dillon model as it required less information and produced the

TABLE 9. NUMBER OF U.S. LAKES SAMPLED BY NES CLASSIFIED AS OLIGOTROPHIC, MESOTROPHIC, AND EUTROPHIC BY SUMMER CHLA AND TP CRITERIA

Classified by TP Criteria*	Classified by CHLA Criteria†			
	Oligotrophic	Mesotrophic	Eutrophic	Total
Oligotrophic	19‡	18↑	2↑	59
Mesotrophic	19↓	45‡	33↑	97
Eutrophic	40↓	134↓	447‡	621
Total	78	197	482	757

*TP criteria are <10 µg/l TP = oligotrophic, 10 > µg/l TP <20 = mesotrophic, and ≥20 µg/l TP = eutrophic.
†CHLA criteria as <2.3 µg/l CHLA = oligotrophic, 2.3 ≤ µg/l CHLA <6.4 = mesotrophic, and ≥6.4 µg/l CHLA = eutrophic.
‡Lakes which are classified trophically the same by both the CHLA and TP criteria.
↑Lakes which are classified trophically higher (more eutrophic) by the CHLA criteria than by the TP criteria.
↓Lakes which are classified trophically lower (less eutrophic) by the CHLA criteria than by the TP criteria.

highest Spearman rank correlation coefficient in the comparison study. Therefore, only the application of the RA to the Larsen/Mercier model is presented in this report. However, the same concept can be easily used to modify any model which predicts ambient TP concentration by modifying the TP levels used to divide lakes into trophic categories according to the lake's RA. Likewise, if the commonly accepted convention of using TP criteria of 10 and 20 µg/l is not suitable for a worker's purpose, the same concept can be used to modify models based on other TP and CHLA criteria for trophic classification.

The concept of modifying loading and ambient models to predict trophic state was developed from a data base which contains only one summer sampling round per lake. Because of the large population of lakes represented in our data base we believe that the apparent trends in our data represent the actual situations even though data for an individual lake may not be precise. We suggest that water sampling be conducted intensively enough to accurately estimate ambient summer CHLA and TP concentrations before a lake's observed RA is used to modify loading or ambient models which predict trophic state.

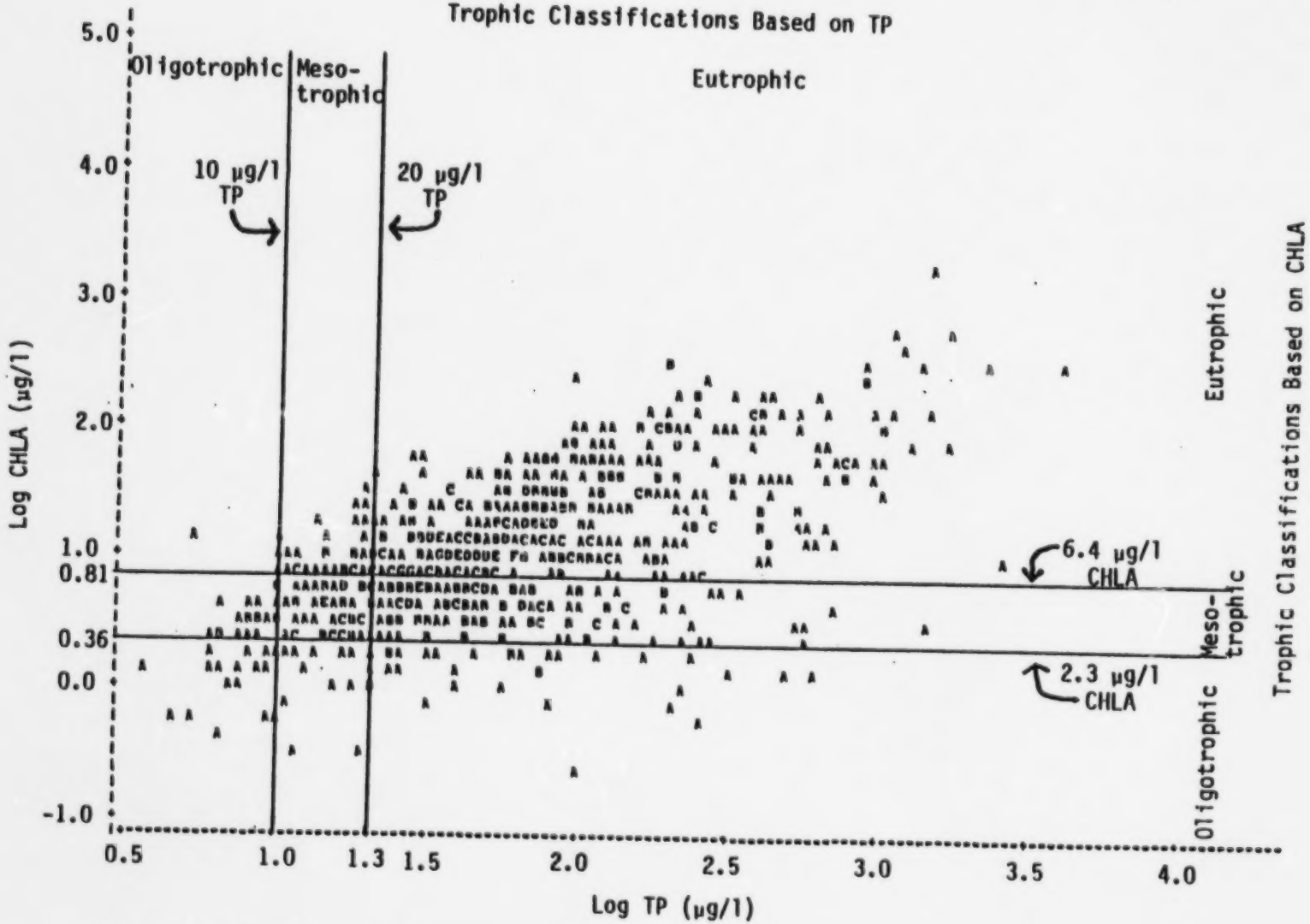

Figure 8. Trophic classifications based on summer TP and CHLA criteria. TP criteria are: <10 µg/l TP = oligotrophic, 10 ≥ µg/l TP <20 = mesotrophic, and >20 µg/l TP = eutrophic. CHLA criteria are: <2.3 µg/l CHLA = oligotrophic, 2.3 ≥ µg/l CHLA <6.4 = mesotrophic, and ≥6.4 µg/l CHLA = eutrophic.

In summary, it is important to adjust for a lake's RA, when using loading and ambient models which predict trophic state because:

1. non-CHLA light interferences and other interferences significantly decrease most lakes' RA and

2. excessive algal growth or other manifestations of nutrient enrichment are more important from a water quality standpoint than a trophic classification based on an arbitrary ambient TP level.

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